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Effect of tile effluent on nutrient concentration and retention efficiency in agricultural drainage ditches

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ABSTRACT

Tile drainage is a common water management practice in many agricultural landscapes in the Midwestern United States. Drainage ditches regularly receive water from agricultural fields through these tile drains. This field-scale study was conducted to determine the impact of tile discharge on ambient nutrient concentration, nutrient retention and transport in drainage ditches. Grab water samples were collected during three flow regimes for the determination of soluble phosphorus (SP), ammonium nitrogen (NH_4^+-N), nitrate nitrogen (NO_3-N) concentrations and their retention in three drainage ditches. Measured nutrient concentration indicated lower SP and NH_4^+-N, and greater NO_3-N concentrations in tile effluents compared to the ditch water. Net uptake lengths were relatively long, especially for NO_3-N, indicating that nutrients were generally not assimilated efficiently in these drainage systems. Results also indicated that the study reaches were very dynamic showing alternating increases or decreases in nutrient concentration across the flow regimes. The drainage ditches appeared to be nutrient-rich streams that could potentially influence the quality of downstream waters.

1. Introduction

Subsurface drainage or tile drainage is a commonly used form of water management in regions that experience high water table during the growing season (Kladivko et al., 2004). Tile drains serve as water transport pathways to receiving drainage ditches. These ditches are essentially headwater streams which act as direct links between agricultural fields and naturally occurring streams or rivers (Smith and Pappas, 2007; Ahiablame et al., 2010).

Although drainage ditches play a significant role in agricultural production, they can also transport nutrients and other agricultural pollutants to downstream waters (Randall and Vetsch, 2005; Needelman et al., 2007; Smith et al., 2008). Studies have, therefore, linked drainage ditches to the hypoxia in the Gulf of Mexico, and to eutrophication in the Great Lakes (Richards et al., 2002; Petrolia and Gowda, 2006; Alexander et al., 2008). Agricultural streams in Indiana and other Midwestern states (e.g. Iowa, Illinois, Ohio) are reportedly major contributors of NO_3-N to the Mississippi River (Goolsby et al., 2001; Mitsch et al., 2001; Royer et al., 2006). Consequently, it is common to see NO_3-N concentration in headwater streams exceeding 10 mg L^{-1} in the Midwest (David et al., 1997; Goolsby et al., 1999). Indiana is also home of some the most fertile mollisols and alfisols in the world with intensive agricultural practices (Richards et al., 2002), and extensive networks of subsurface tile drainage.

While drainage ditches may increase delivery of NO_3-N to streams, they have the potential to mitigate sediment and phosphorus (P) losses from agricultural landscapes (Kröger et al., 2008). This attenuation capacity has prompted interest in identifying drainage management strategies for decreasing P transport in drainage ditches (Smith et al., 2006a,b; Smith and Pappas, 2007). Despite these efforts, P losses to drainage ditches are still a matter of concern (Smith et al., 2006a,b; Smith and Huang, 2010). Studies related to subsurface drainage have been conducted for many years (Skaggs and Chescheir, 2003; Kladivko et al., 2004); however, the topic has recently received renewed attention due to increased concerns over the hypoxic zone in the Gulf of Mexico (Sharpley et al., 2007; Strock et al., 2007).

A nutrient molecule, after entering receiving waters, travels from dissolved form to particulate form, and back to dissolved form (Davis and Minshall, 1999; Haggard et al., 2001; Tank et al., 2006). The distance the molecule travels in dissolved before it is assimilated in particulate form is referred to as net uptake length, S_w (Stream Solute Workshop, 1990; Haggard et al., 2001; Wollheim

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et al., 2001); and the distance the molecule travels in particulate form, before being released back to the water column, is the turnover length, S_s (Stream Solute Workshop, 1990). The combination of uptake and turnover lengths constitutes a complete cycle, and has been described as nutrient spiraling length, S (Davis and Minshall, 1999; Haggard et al., 2001; Chaubey et al., 2007). Spiraling length is a measure of nutrient retention capacity of a stream (Newbold, 1992). The occurrence of these cycles is regulated by the spiraling length and the downstream movement of water; a process known as nutrient dynamics or nutrient cycling (Stream Solute Workshop, 1990; Haggard et al., 2001). The expression nutrient/solute dynamics denotes the spatial and temporal patterns and transport of materials that are chemically dissolved in water (Stream Solute Workshop, 1990).

Nutrient spiraling is generally quantified using a metric triad (Webster and Valett, 2006) consisting of uptake length, uptake rate (U), and uptake velocity or mass transfer coefficient (V_f). A short spiraling length generally indicates a relatively healthy stream, whereas a long spiraling length implies a nutrient rich stream with relatively low intrinsic ability to assimilate nutrients. Previous studies have reported variations in the spiraling length with space, time and season. Temporal and seasonal variations depend mostly on temperature, flow regimes, stream communities and allochthonous deposits in the stream (Webster et al., 1991; Haggard et al., 2001; Doyle et al., 2003). Spatial variations may be influenced by channel size, upland activities, and environmental conditions in the watershed (D'Angelo et al., 1991; Marti and Sabater, 1996; Haggard et al., 2001).

Nutrient losses from agricultural fields to drainage ditches have been identified as a major contributor to downstream water quality deterioration. However, more documentation on nutrient retention in drainage ditches receiving inputs from tile drains is needed to improve drainage management strategies. The goal of this study was to examine nutrient concentrations in tile effluent and their resulting dynamics in agricultural drainage ditches. The specific objectives were to examine: (1) effect of tile effluent on nutrient concentration in the ditch water at tile outlets and downstream locations; (2) nutrient retention efficiency in three tile-fed drainage ditches in Indiana.

2. Materials and methods

2.1. Site description

This study was conducted in three tile-fed drainage ditches in Northwest Indiana. The ditches, which generally flow in a straight line, are representative of typical Midwest drainage ditches (Fig. 1). The J.B. Foltz Ditch is located near Reynolds, Indiana, and acts as one of the headwater streams (Fig. 2a) discharging in the Minch Ditch, Hoagland Ditch, and Tippecanoe River. The J.B. Foltz Ditch drains approximately 8 km^2 out of the 182 km^2 of the entire area of the Hoagland Ditch watershed. The Hoagland Ditch watershed is located in Benton, Jasper and White Counties in northwest Indiana. The reach of the ditch used for this study, which drains about 2 km^2, is mostly vegetated and has not been dredged for many years. The common vegetation observable in the riparian area of the ditch during summer includes big bluestem (*Andropogon gerardii*), Indian grass (*Sorghastrum*), and switchgrass (*Panicum virgatum*) with a mean height of 1.3 m. Ninety three percent of the land area contributing flow to the ditch is heavily farmed with corn–soybean rotation under no-till row crop. The remaining 7% is covered by a combination of low density residential area, pasture and grass. Predominant soils in the contributing area of the study reach are Gilford sandy loam (coarse-loamy, mixed, superactive, mesic Typic Endoaquolls) and Rensselaer loam (fine-loamy, mixed, superactive, mesic Typic Argiaquolls) with an average slope less than 0.6%.

Fig. 1. Photograph of Marshall Ditch during a sampling event.

The other two study ditches are Box Ditch and Marshall Ditch located near West Lafayette, Tippecanoe County, Indiana in Little Pine Creek watershed (Fig. 2b). Both ditches are headwater streams draining approximately 8 km^2 each out of the 53 km^2 of the entire Little Pine Creek watershed. The land use is primarily agricultural and livestock production (90%) and low density residential areas (10%). Corn–soybean rotation with no-till is the main cropping system in the watershed areas contributing flow to the ditches. Each study reach of the ditches drains approximately 1 km^2 of Drummer silty clay loam (fine-silty, mixed, superactive, mesic Typic Endoaquolls) and Toronto silt loam (fine-silty, mixed, superactive, mesic Typic Endoaquolls) prevalent in the watershed. Toronto soils are more abundant in the Box Ditch. The riparian area on the study reaches is favorable to tall grass, consisting of big bluestem, Indian grass, and switchgrass during the summer. The average slope in the area drained by the ditches varies between 0 and 2%. Portions of agricultural fields located in the Marshall Ditch watershed are periodically irrigated with effluent from a swine lagoon located in the watershed.

2.2. Field techniques

Tile drains are designed to convey water from the fields into the receiving ditches and tile outlets generally discharge at regular spatial intervals. However, some of the studied ditches have a limited number of tile outlets. To have a relatively equal number of outlets and a reasonable study reach for the three ditches, a distance of more than 200 m between two outlets was used as criteria to select the tile outlets. There are 5 outlets in the J.B. Foltz Ditch, 5 in the Box Ditch, and 4 in the Marshall Ditch (Table 1). The first two outlets in Box Ditch were considered as one outlet due to the short distance between them (Table 1). Water samples were collected by grab on 9 dates from February to July 2008 (n = 9 per ditch). Samples were collected directly from tile drain outflow (herein referred to as tile discharge or tile effluent), and approximately 5 m upstream and downstream from each selected outlet. Samples were also collected between outlets at specific transects (up to 5) spaced at least 30 m apart, when the distance between tile outlets allowed multiple transects on the study reaches. Table 2 shows the number of transects established on each study reach. It should be noted that only 1 transect between outlets 3 and 4 and 0 transect between outlets 4 and 5 were established in Box Ditch due to safety reasons.

About 60 mL of filtered water was collected at each sampling point during each of the 9 events. Samples were first collected using 500 mL nalgene bottles from the middle of the ditch, before

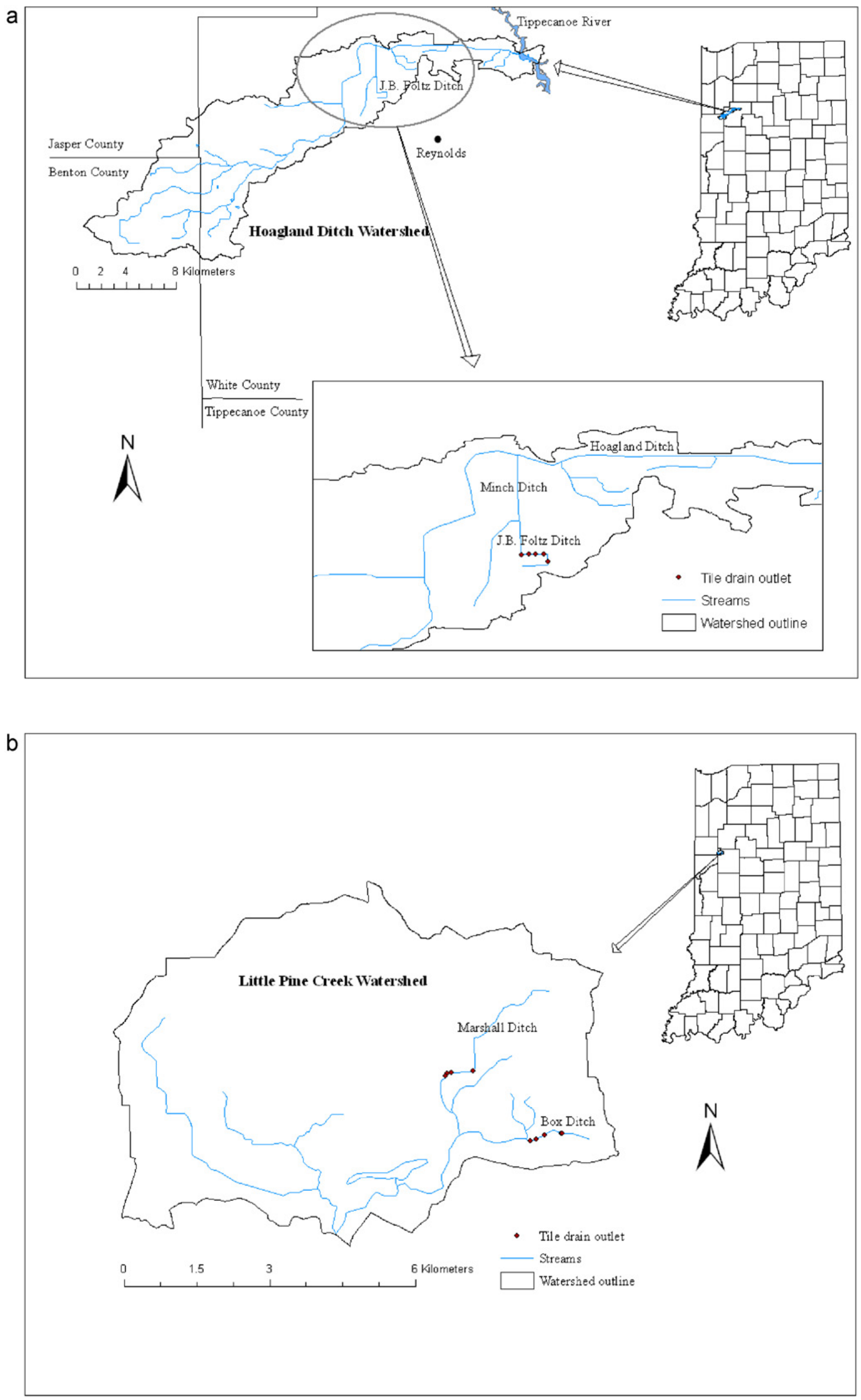

Fig. 2. (a) Location of Hoagland Ditch watershed, J.B. Foltz Ditch, and Sampling Sites in Indiana. (b) Location of Little Pine Creek watershed, Box Ditch, Marshal Ditch, and sampling sites in Indiana.

being filtered into pre-labeled sterilized 60 mL nalgene bottles. Samples were manually filtered on site using a 30 mL high density polyethylene (HDPE) syringe and a 0.45 μm nylon filter membrane (diameter = 25 mm). All samples were stored on ice in the dark until transported to the laboratory. To minimize variations in flow and stream characteristics, samples of one study reach were collected within 1.5 h. After returning to the laboratory, samples were preserved at pH <2 with H_2SO_4 and stored at 4 °C until analyzed.

Table 1
Study sites showing global positioning system coordinates of selected tile drain outlets and respective distances from the most upstream tile outlet within each reach.

Ditch	Tile outlet	Diameter (cm)	Longitude	Latitude	Distance (m)
J.B. Foltz Ditch, Reynolds, IN	1	17	W86°55.215	N40°46.868	0
	2	15	W86°55.295	N40°47.020	385
	3	20	W86°55.464	N40°47.017	621
	4	20	W86°55.608	N40°47.010	826
	5	20	W86°55.756	N40°47.006	1036
Box Ditch, West Lafayette, IN	1[a]	183	W86°59.899	N40°29.659	0
	2	23	W86°59.897	N40°29.658	7.3
	3	23	W87°00.114	N40°29.668	336
	4	23	W87°00.269	N40°29.593	456
	5	23	W87°00.352	N40°29.572	523
Marshall Ditch, West Lafayette, IN	1	36	W87°01.186	N40°30.351	0
	2[b]	31	W87°01.506	N40°30.336	431
	3	23	W87°01.564	N40°30.328	499
	4	36	W87°01.588	N40°30.297	558

[a] 1: small ditch entering the main ditch through a circular pipe.
[b] 2: conventional weir flow control structure with circular spillway.

Table 2
Established transects between tile drain outlets for water quality sampling.

Transect	J.B. Foltz	Box	Marshall
T_{1-2}	5	0	5
T_{2-3}	5	5	1
T_{3-4}	5	1	1
T_{4-5}	5	0	0

T_{1-5}: transect between two tile drain outlets.

During each sampling event, water depth, width, and velocity were measured using an electromagnetic Flow Tracker (SonTek Handheld ADV®). Measurements of flow velocity taken at several cross-sections between selected outlets were used to calculate the average discharge utilized in net uptake length calculations. Conductivity, temperature, and dissolved oxygen were also measured using a conductivity meter 115A plus (YSI Model 85, Yellow Springs, OH); pH was measured with a Chek-Mite pH-15 Sensor. Flow rates of tile drain effluents were measured using the bucket method. The bucket method is a simple, low technology method for determining flow in small streams and rivers using a stopwatch and a bucket or a container. In this study the time it took for tile effluents to fill a 15-L bucket was recorded six times and the average calculated for each tile drain outlet. This technique, when permitted by the study reach, was used to measure flow rate of tile effluents at all selected tile outlets and during all sampling events.

2.3. Classification of flow regime

Continuous daily flow data for the period covering the 9 sampling events ($n = 9$ per ditch) was collected from monitoring stations installed in the J.B. Foltz Ditch and the Marshall Ditch. This information was used as a guide to classify flow regimes in the three ditches. The probability of exceedance derived from the flow duration curve (FDC), developed for each ditch, was utilized to classify the flow regimes. Each flow regime is a description of probability that a given amount of streamflow was equaled or exceeded (Table 3). The 9 sampling events were grouped into low, medium or intermediate, and high flow regimes. The high flow regime was described by all flows with less than or equal to 10% probability of occurrence. The medium flow regime corresponded to flows not exceeding 10–55% of the time; and the low flow regime ranged between 55 and 100% flow exceedance range. For example, there was 0–10% probability that flows in J.B. Foltz Ditch will be equal to or greater than 0.9 L s^{-1} on February 9, March 20, and March 29, 2008.

2.4. Laboratory techniques and data analysis

Samples were analyzed for Soluble P (SP), ammonium nitrogen (NH_4^+-N), nitrate nitrogen (NO_3-N) and chloride (Cl^-). SP concentrations were determined using the inductively coupled plasma-optical emission spectrometry (ICP-OES, Optima 2000 DV, Norwalk, CT). The determination of SP with this protocol includes other forms of P such as polyphosphates, colloidal P, and dissolved hydrolysable P filterable through 0.45 μm membranes (Rowland and Haygarth, 1997; Jarvie et al., 2002). NH_4^+-N was analyzed by the reaction of alkaline phenate with hypochlorite and sodium nitroprusside (indophenol blue) method (EPA-103-A; Analyzer AQ2$^+$, Milwaukee, WI). NO_3-N was analyzed by the cadmium coil reduction method in a reaction of sulfanilamide, phosphoric acid and N-(1-naphthyl)-ethylenedimine dihydrochloride (EPA-114-A; Analyzer AQ2$^+$, Milwaukee, WI). The cadmium reduction method combines nitrate + nitrite (NO_2-N) nitrogen fractions as NO_3-N in the end reports. The fractions of NO_2-N alone can be determined with a separate laboratory matrix; but NO_2-N portions are usually very small, and assumed negligible in this study. Cl^- concentra-

Table 3
Flow (L s^{-1}) regime in the three ditches.

	Prob. of exceedance								
	0–10%			10–55%			55–100%		
	9 February	20 March	29 March	16 May	2 June	14 July	9 May	23 June	21 July
J.B. Foltz Ditch	1.0	1.0	0.9	0.4	0.4	0.4	0.04	0.1	0.05

	Prob. of exceedance								
	0–10%			10–55%			55–100%		
	11 April	16 May	2 June	9 February	29 March	9 May	20 March	14 July	21 July
Box and Marshall Ditches	262	426	1027	21	43	32	0.2	9.0	2.0

tions were determined using the mercuric thiocyanate reaction with ferric nitrate method (EPA-105-A; Analyzer AQ2$^+$, Milwaukee, WI).

Previous studies have used Cl^- as a hydrologic tracer to correct for longitudinal dilution and background concentration for nutrients (Haggard et al., 2001; Marti et al., 2004; Chaubey et al., 2007). This study did not correct either for background concentration or dilution from lateral and subsurface inflow because no substantial decrease in Cl^- concentrations between the outlets was observed. Net nutrient uptake lengths were the result of natural decline in nutrient concentrations in the ditches (Haggard et al., 2001). Net uptake length, S_{net}, was calculated using the inverse slope of downstream decline in nutrient concentrations (Stream Solute Workshop, 1990; Doyle et al., 2003; Webster and Valett, 2006) as:

$$[C]_x = [C]_o\, e^{-kx} \tag{1}$$

$$S_{net} = \frac{-1}{k} \tag{2}$$

where $[C]_o$ is the concentration of nutrient at the station immediately below a tile outlet, which, in this case, is the station 5 m downstream from a tile outlet; $[C]_x$ is the concentration of nutrient at downstream stations; and k is the coefficient of downstream decline of nutrient (m^{-1}). To account for input from each tile outlet along the study reaches, S_{net} was calculated for SP, $NH_4{}^+$-N and NO_3-N in the three ditches for each segment between two outlets. The mean S_{net} for an entire study reach was determined by normalizing all partial S_{net} with the length of reach segments. The average S_{net} can be mathematically expressed as:

$$S_{net} = \sum \frac{S_{net(i)} x_i}{x_{total}} \tag{3}$$

where S_{net} is the normalized mean net uptake length of the entire study reach; $S_{net(i)}$ is the net uptake length of segment (i) between two outlets; x_i is the longitudinal distance between two outlets; and x_{total} is the length of the entire study reach. Mass transfer coefficients or uptake velocity, V_f (mm min^{-1}), which describes how fast nutrient molecules in the water column move towards sediments (Doyle et al., 2003), were also calculated using the normalized S_{net} (Davis and Minshall, 1999; Haggard et al., 2001):

$$V_f = \frac{vh}{S_{net}} \tag{4}$$

where v is the average water velocity (m s^{-1}), and h is the average water depth of the study reach (m). Nutrient loadings to the ditches through tile outlets were estimated using:

$$L_n = [C]Q_t \tag{5}$$

where L_n is the amount of nutrient passing through a tile outlet per unit time (mg s^{-1}); $[C]$ is nutrient concentration per unit volume of water (mg L^{-1}); and Q_t is the discharge of a tile outlet (L s^{-1}).

2.5. Statistical analysis

To assure a normal distribution of variables, a natural logarithmic transformation is generally performed prior to analyzing water quality data (Hirsh et al., 1991; Haggard et al., 2001; Ekka et al., 2006). Thus, all variables in this study were log-transformed. Differences in water quality of tile effluent and ditch water were evaluated using a two-way repeated measures analysis of variance (ANOVA) with the significance level set at $\alpha = 0.10$. The Statistical Analysis System package, version 9.1, (SAS Institute Inc., 2003) was used for all analyses.

3. Results and discussion

3.1. Ditch discharge, temperature, dissolved oxygen, pH, and conductivity

The discharge calculated from measured flow velocity during sampling events varied among ditches and sampling events. J.B. Foltz Ditch, the widest and deepest ditch, had the lowest discharge ranging between 0.005 and 0.120 m^3 s^{-1}. Box Ditch and Marshall Ditch are located in the same watershed; however, average discharge was greater in Box Ditch (0.001–0.800 m^3 s^{-1}) than in Marshall Ditch (0.003–0.250 m^3 s^{-1}). The average temperature in the three ditches did not show any abnormality, and was consistent with seasonal temperature variations in natural waters of cold regions. These seasonal temperature variations, however, influenced dissolved oxygen (DO) during warmer months. DO varied between 2.1 mg L^{-1} in warmer months and 8.5 mg L^{-1} in colder months in the three ditches. Decreased DO levels recorded during warmer months could be the direct result of limited oxygen holding capacity caused by the temperature rise in these ditches. Unlike the discharge, the pH did not change much across sampling events. With a range that varied between 7.3 and 8.3, the average pH in these ditches fell within the range of 6.5–9.0 published in the Indiana Administrative Code 2 (2008). During the sampling period, conductivity varied within each ditch and across ditches (range, 82–850 μS). High conductivity was observed during summer months, whereas lower conductivity occurred during winter months. Seasonal high discharge from tile drains during winter months owing to rainfall-runoff and snow melt could result in dilution which may lower conductivity levels.

3.2. Nutrient inputs from tile drains

Tile outflows recorded during each sampling event were used to estimate the amount of nutrients entering these ditches through the selected tile drains (Eq. (5)). During the study period, tile outlets in the J.B. Foltz Ditch were partially or totally submerged, resulting in direct interactions between the ditch water and tile effluents. Consequently, estimation of nutrient inputs from tile effluents in the J.B. Foltz Ditch was not possible with the tile discharge measuring technique used in this study.

Even though tile effluents varied across sampling events and sites, considerable amounts of nutrients were lost to the ditch water in Box Ditch and Marshall Ditch (Table 4). SP loading in Box ditch and Marshall Ditch ranged from 0.0003 to 2.6 mg s^{-1} with highest loads observed in Marshall Ditch. NO_3-N loading was higher than 10 mg s^{-1} during 4 sampling events in Box Ditch, and 7 sampling events in Marshall Ditch. Higher SP and NO_3-N loading in Marshall Ditch could be the result of manure irrigation in the surrounding fields. While there is no consistent pattern across flow regimes (low–medium–high) for SP and $NH_4{}^+$-N loading, NO_3-N and Cl^- loads in the two ditches tended to increase with increase in flow conditions (Table 4). Land use in fields draining to the study drainage ditches is representative of land use in agricultural watersheds in Indiana. Thus, the estimated nutrient loading generally reflects the amount of nutrients passing through tile drains into agricultural drainage ditches from this region.

3.3. Quality of tile effluent and ditch water

Nutrient concentrations in tile effluents were averaged for each sampling date, and nutrient concentrations measured in the ditch water (at the various transects) along the study reaches on each sampling date were also averaged to determine mean nutrient concentrations in tile effluent and in the ditch water. Average SP concentrations, calculated based on the summation of P concen-

Table 4
Estimated nutrient discharge in tile effluent in Box Ditch and Marshall Ditch during all sampling events.

	Q_t ($L\,s^{-1}$)	SP ($mg\,s^{-1}$)	NH_4^+-N ($mg\,s^{-1}$)	NO_3-N ($mg\,s^{-1}$)	Cl^- ($mg\,s^{-1}$)
Box					
9 February	0.6	0.0003	0.0	8.9	10.8
20 March	0.2	0.01	0.01	2.1	0.5
29 March	0.2	0.001	0.01	3.2	2.8
11 April	2.7	0.02	0.10	39.5	40.8
9 May	0.3	0.003	0.01	3.3	4.6
16 May	3.7	0.6	0.5	51.6	32.6
2 June	10.0	0.1	0.04	188	163
14 July	1.2	0.03	0.02	15.7	17.1
21 July	0.8	0.03	0.02	6.1	12.1
Marshall					
9 February	3.5	2.6	1.4	8.5	20.9
20 March	0.6	0.3	1.8	7.6	3.5
29 March	2.3	0.1	0.2	26.5	77.4
11 April	3.8	0.6	3.2	50.3	87.6
9 May	6.0	0.0	0.2	101	204
16 May	6.5	0.9	1.3	156	143
2 June	6.8	0.1	0.03	186	208
14 July	5.5	0.04	0.1	94.0	173
21 July	2.4	0.06	0.05	33.4	64.3

trations in tile effluent and the ditch water, ranged from 0.011 to 0.211 $mg\,L^{-1}$, 0.009 to 0.186 $mg\,L^{-1}$, and 0.004 to 0.264 $mg\,L^{-1}$ in the J.B. Foltz Ditch, the Box Ditch and the Marshall Ditch. These values were consistent with the range of 0.020–0.120 $mg\,L^{-1}$ for drainage ditches from this region (Smith et al., 2006b). Although SP concentrations in tile effluent were significantly lower in the J.B. Foltz Ditch and in the Box Ditch ($p < 0.04$), SP in the Marshall Ditch did not exhibit the same pattern (Table 5). Analysis of the pooled data from the three sites showed a significant influence of tile effluent on the ditch water ($r^2 > 0.85$; $p < 0.0003$). Previous studies reported reduced P losses from subsurface drainage systems (Gilliam et al., 1999; Kladivko et al., 2004). P has a high affinity for soil particles such that during rainfall and snow melt events, P may enter waterways in the form of particulate and dissolved P through surface flow and surface runoff (Sharpley et al., 1994; Kröger et al., 2008). The significance of the impact of tile effluent when the data was pooled together may be the influence of the Marshall Ditch data. Many fields in the Marshall Ditch watershed are periodically irrigated with water from animal waste lagoons, and high P loading in this ditch could be the result of elevated P leaching into subsurface tile drains from the watershed. Moreover, grazing is a common practice upstream from the Marshall Ditch site, contributing to the likelihood of greater P loading in tile drains on the study reach.

Average NH_4^+-N concentrations varied between 0.023 and 0.148 $mg\,L^{-1}$ in J.B. Foltz Ditch, 0.007 and 0.216 $mg\,L^{-1}$ in Box Ditch, and 0.007 and 0.689 $mg\,L^{-1}$ in Marshall Ditch. Tile effluents impacted ditch water NH_4^+-N in all the ditches except in Box Ditch ($p < 0.07$).

Average NO_3-N concentrations ranged from 1.11 to 7.99 $mg\,L^{-1}$ in J.B. Foltz Ditch, from 6.5 to 16.1 $mg\,L^{-1}$ in Box Ditch, and from 9.3 to 23.8 $mg\,L^{-1}$ in Marshall Ditch. Unlike SP, tile effluents significantly impacted ditch water NO_3-N in Box Ditch and J.B. Foltz Ditch, along with Marshall Ditch ($p < 0.005$). Collective effects of tile effluents of the three ditches resulted also in significant increase of NO_3-N concentrations in the ditch water due to elevated NO_3-N discharge from tile drains ($r^2 > 0.94$; $p < 0.0001$).

Nutrient concentrations in the Marshall Ditch were generally greater than nutrient concentrations in the other two ditches (Table 5). The study compared well with previous work reported on other Midwest drainage ditches (Randall and Vetsch, 2005; Gentry et al., 2007; Strock et al., 2007). For example, SP concentrations were comparable to the range of 0.001 and 0.5 $mg\,L^{-1}$ reported for northeast Indiana ditches (Smith et al., 2008). NH_4^+-N concentrations were comparable to the range of 0.01 and 1.3 $mg\,L^{-1}$ reported from Indiana (Smith et al., 2008), and 0.01 and 2.6 $mg\,L^{-1}$ reported from Illinois (Gentry et al., 2000). NO_3-N was also consistent with the reported range of 0.09 and 19.2 $mg\,L^{-1}$ for Indiana drainage ditches (Smith et al., 2008), and 8.3 and 14.9 $mg\,L^{-1}$ from tile drained fields in Illinois (Gentry et al., 2000). In Minnesota, drainage studies showed that NO_3-N averaged 28 $mg\,L^{-1}$ (Randall et al., 2007), and a range varying between 8.6 and 29.3 $mg\,L^{-1}$ was observed in Iowa drainage systems (Jaynes et al., 1999). Similar studies in Ohio and Illinois indicated that subsurface drainage water may contain high NO_3-N concentrations from agricultural fields (Logan et al., 1994; Royer et al., 2004). Results from comparisons between tile effluent and ditch water nutrient concentrations confirmed that tile effluent is major transport pathways of nutrients, especially NO_3-N, to the receiving ditches.

3.4. Nutrient retention

Longitudinal patterns observed in nutrient concentrations within each ditch showed increases or decreases between tile outlets, but were consistent with patterns in Cl^- concentrations. Significant uptake for SP, in J.B. Foltz Ditch (Fig. 3), occurred only on May 16 and with a corresponding S_{net} of 22,700 m, and V_f of 0.46 $mm\,min^{-1}$ ($p < 0.09$). During the sampling period, NH_4^+-N uptake was significant on 2 dates (Fig. 3). The S_{net} for NH_4^+-N ranged

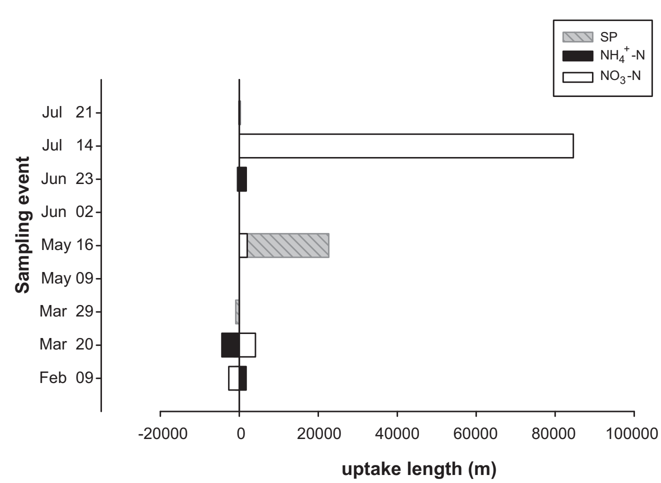

Fig. 3. Mean nutrient uptake length for J.B. Foltz Ditch by sampling date.

Table 5
Mean of nutrient concentration (mg L^{-1}) in the ditch water and in tile effluent at the three sites.

	Sample type	SP	NH_4^+-N	NO_3-N	Cl^-
J.B. Foltz					
9 February	Tile effluent	0.13	0.07	2.29	2.32
	Ditch water	0.13	0.08	0.92	1.91
20 March	Tile effluent	0.02	0.03	9.27	17.33
	Ditch water	0.01	0.02	3.12	15.33
29 March	Tile effluent	0.02	0.05	13.53	15.49
	Ditch water	0.01	0.05	4.16	12.96
11 April	Tile effluent	0.01	0.06	2.26	14.98
	Ditch water	0.02	0.08	1.23	14.23
9 May	Tile effluent	0.14	0.03	7.95	17.05
	Ditch water	0.22	0.04	3.46	16.84
16 May	Tile effluent	0.03	0.03	7.68	17.27
	Ditch water	0.03	0.02	7.27	17.47
2 June	Tile effluent	0.01	0.10	7.86	18.02
	Ditch water	0.02	0.16	3.82	17.39
14 July	Tile effluent	0.01	0.05	5.73	12.57
	Ditch water	0.01	0.04	4.68	12.86
21 July	Tile effluent	0.04	0.24	26.65	16.07
	Ditch water	0.08	0.25	4.88	14.97
Box					
9 February	Tile effluent	0.00	0.00	14.79	17.94
	Ditch water	0.06	0.03	8.43	10.48
20 March	Tile effluent	0.03	0.08	11.69	2.81
	Ditch water	0.02	0.08	8.76	2.90
29 March	Tile effluent	0.00	0.03	14.39	12.64
	Ditch water	0.02	0.04	10.36	13.03
11 April	Tile effluent	0.01	0.04	14.58	15.06
	Ditch water	0.02	0.04	11.36	13.93
9 May	Tile effluent	0.01	0.03	11.57	16.11
	Ditch water	0.01	0.02	11.14	18.62
16 May	Tile effluent	0.16	0.14	14.08	8.89
	Ditch water	0.19	0.24	15.95	10.34
2 June	Tile effluent	0.01	0.00	18.80	16.28
	Ditch water	0.06	0.01	15.32	15.79
14 July	Tile effluent	0.03	0.02	13.67	14.85
	Ditch water	0.09	0.02	9.83	13.94
21 July	Tile effluent	0.04	0.03	7.77	15.39
	Ditch water	0.04	0.02	6.14	15.54
Marshall					
9 February	Tile effluent	0.73	1.84	10.75	26.59
	Ditch water	0.14	0.16	9.11	13.19
20 March	Tile effluent	0.46	3.05	12.63	5.85
	Ditch water	0.05	0.06	13.13	3.41
29 March	Tile effluent	0.06	0.10	11.34	33.16
	Ditch water	0.09	0.13	9.93	13.95
11 April	Tile effluent	0.16	0.84	13.12	22.86
	Ditch water	0.01	0.01	15.98	18.37
9 May	Tile effluent	0.00	0.04	16.73	33.79
	Ditch water	0.00	0.04	13.37	19.73
16 May	Tile effluent	0.14	0.21	23.95	21.97
	Ditch water	0.18	0.11	15.74	9.32
2 June	Tile effluent	0.01	0.00	27.24	30.49
	Ditch water	0.01	0.01	22.90	21.76
14 July	Tile effluent	0.01	0.03	17.20	31.72
	Ditch water	0.01	0.04	13.38	17.18
21 July	Tile effluent	0.03	0.02	13.81	26.61
	Ditch water	0.04	0.05	8.12	21.66

from 1726 to 1745 m with corresponding V_f ranging from 1.52 to 3.4 mm min^{-1}. Uptake for NO_3-N occurred on 3 out of 9 events, with S_{net} ranging from 2025 to 84,650 m, and V_f from 0.00039 to 0.12 mm min^{-1} (Fig. 3).

Net uptake lengths for SP ranged from 374 to 12,980 m in Box Ditch, with V_f corresponding to 4.1 and 0.041 mm min^{-1} (Fig. 4). While uptake for NH_4^+-N did not occur in Box Ditch during all sampling events, NO_3-N uptake occurred at least during the March 20 sampling event ($p = 0.07$), with S_{net} of 30,433 m and V_f of 0.023 mm min^{-1} (Fig. 4).

Significant uptake for SP ($p < 0.02$) occurred only on June 2 in Marshall Ditch (Fig. 5). The calculated S_{net} along the study reach was 1190 m and V_f was 1.6 mm min^{-1} for this event. On March 20,

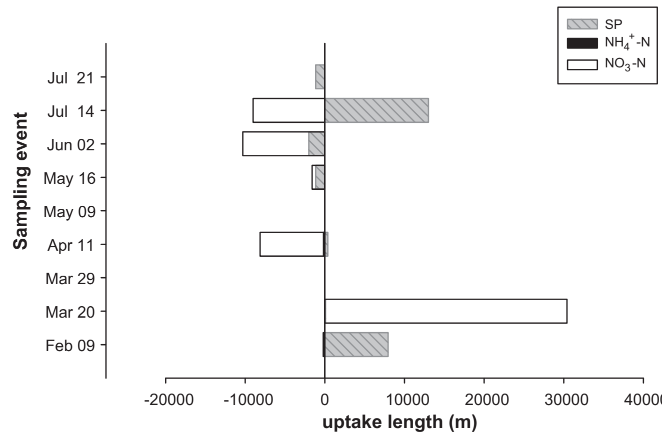

Fig. 4. Mean nutrient uptake length for Box Ditch by sampling date.

Marshall Ditch significantly assimilated NH_4^+-N ($p = 0.09$) with a S_{net} of 1270 m (Fig. 5). The Marshall Ditch exhibited uptake for NO_3-N ($p < 0.08$) during 4 out of 9 sampling events (Fig. 5). The S_{net} for these events varied between 2030 and 6510 m with V_f ranging from 0.074 to 0.37 mm min^{-1}.

Previous studies in agricultural dominated streams in the Midwest estimated S_{net} less than 100–750 m for SP with a corresponding V_f ranging from 1.0 to 5.0 mm min^{-1} (Bernot et al., 2006). Studies from other parts in the USA also reported a range of 1000–31,000 m for SP (Haggard et al., 2001) with V_f ranging from 0.50 to 6.6 mm min^{-1} (Chaubey et al., 2007), and from 1.9 to 12 mm min^{-1} (Hall et al., 2002). In the J.B. Foltz Ditch, S_{net} and V_f for SP were comparable to the estimated range from other parts of the nation. While average S_{net} for SP in Box Ditch was found to be greater than the reported S_{net} range in agriculturally dominated streams from the Midwest region, it was shorter than S_{net} in the other two ditches. This observation suggests that Box Ditch retained P relatively better than the other two ditches during the study period. Likewise, V_f compared well with other published V_f values. Although Marshall Ditch appeared to have higher P inputs, S_{net} values for SP were consistent with the published data. Out of the three ditches, J.B. Foltz Ditch had the longest S_{net} values for SP.

J.B. Foltz Ditch had the longest S_{net} for NO_3-N with a range larger than 750–2300 m observed by other researchers (Bernot et al., 2006). However, V_f was comparable to the range of 0.50–5.0 mm min^{-1} from the region (Bernot et al., 2006). S_{net} and V_f for NH_4^+-N and NO_3-N in J.B. Foltz Ditch were also similar to

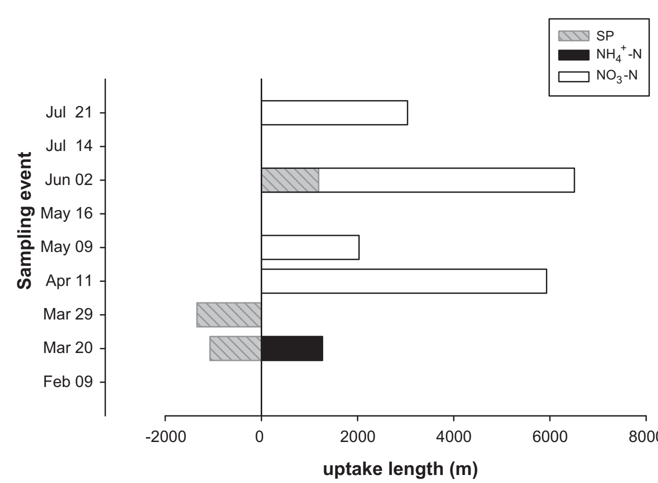

Fig. 5. Mean nutrient uptake length for Marshall Ditch by sampling date.

S_{net} and V_f in Box Ditch. While S_{net} for NH_4^+-N in J.B. Foltz Ditch was longer than the published range of 150–800 m (Bernot et al., 2006), and 760–1350 m (Hamilton et al., 2001), these results did not seem unusual for nutrient-rich agricultural streams. Average S_{net} for NO_3-N in Box Ditch was greater than S_{net} values reported by Bernot et al. (2006), but comparable to the other two ditches and the published data mentioned above. V_f values for NO_3-N in Box Ditch were lower than the range in the literature. Average S_{net} for NH_4^+-N and NO_3-N showed that Marshall Ditch assimilated NH_4^+-N and NO_3-N better than the other two ditches. Nutrient uptake did not appear to have a consistent pattern across the ditches and flow conditions.

3.5. Drainage water quality implications

Over the course of the study, little uptake of nutrients occurred in the ditches, indicating that tile effluents impacted water quality in the three ditches. For example, uptake of P occurred on 1 out of 9 sampling events on average in the three ditches, and P net uptake lengths were quite long. Similarly, uptake of NH_4^+-N occurred only on a few sampling events with relatively long net uptake lengths across the ditches. Even though previous studies have reported that drainage ditches may be used as mitigation tool for P reduction (Kröger et al., 2008), S_{net} for P in this study shows that P was being transported downstream with little retention. Inputs of P from tile drains, even in small amounts (Gilliam et al., 1999; Kladivko et al., 2004), would result in greater P concentrations in the ditch water over time and would more likely alter P retention in the studied drainage ditches. While retention of NH_4^+-N may be due to transformation rather than assimilation (Chaubey et al., 2007), long S_{net} for NH_4^+-N also indicates the inability of the ditches to efficiently remove NH_4^+-N loads.

The presence of high NO_3-N contents in drainage ditches is related to NO_3-N inputs from tile drains (Gentry et al., 2000; Kladivko et al., 2004; Randall et al., 2007). Although net uptake lengths for NO_3-N varied among drainage ditches and between sampling events, they are all in kilometers, indicating significant downstream transport of NO_3-N in the three drainage ditches (Figs. 3–5). Nutrient retention efficiency is affected by many factors such as discharge and velocity (D'Angelo and Webster, 1991; Haggard et al., 2001). However, focusing solely on tile effluents, increased NO_3-N inputs from tile drains would result in increased transport of NO_3-N through the ditches with the potential to conceal biotic and abiotic processes that govern nutrient removal in streams (Stream Solute Workshop, 1990). Transport of NO_3-N over long distances from these headwater agricultural streams would affect the integrity of ecosystems further downstream. Royer et al. (2004) have demonstrated that NO_3-N in these headwater streams are transported, rather than denitrified, to downstream waters, contributing considerable N inputs to the Gulf of Mexico. The study complements the importance of headwater streams on downstream water quality (Alexander et al., 2008).

Even though little uptake of nutrients was observed in these ditches, relatively greater amounts of NH_4^+-N were retained followed by P and NO_3-N in the three ditches. NO_3-N appears to be a non-limiting nutrient in these ditches and would be expectedly transported more abundantly downstream compared to other nutrient species. Samples were collected from February to July, a period in which soils are commonly vulnerable to leaching due to excess rainfall in spring and fall (Keeney and Follett, 1991), allowing increased losses of NO_3-N to the receiving ditches. Previous studies have reported that 75% of annual NO_3-N losses occur in April, May, and June (Randall and Vetsch, 2005). Royer et al. (2006) also reported these seasonal losses of NO_3-N from mid January through June. Agricultural drainage ditches could significantly contribute to NO_3-N loads in the Mississippi River Basin. Results from this study suggest that nutrient inputs from tile drains, even in small amounts, would result in greater nutrient concentrations in the receiving ditches over time, exceeding the intrinsic nutrient retention capacity of the ditches. Consequently, some drainage ditches may not be able to efficiently retain nutrient loads from tile discharge.

Results indicate that nutrients in these drainage ditches were generally not assimilated efficiently. Therefore, natural stream principles should be followed when designing drainage ditches (Evans et al., 2007; Needelman et al., 2007), and management strategies (Faria et al., unpublished data; Penn et al., 2007; Chun et al., 2009; Kresge and Mamen, 2009) should account for tile drain outlets for better reduction of agricultural nonpoint source pollution.

4. Conclusions

This study indicates that agricultural drainage ditches in Northwest Indiana are nutrient-rich streams, especially in N due to elevated NO_3-N inputs from tile drains. Although ambient concentration of P and N in the study ditches compared well with the data from previous studies on Indiana drainage ditches, the capacity of the ditches to retain nutrients was exceeded by these concentrations. Consequently, the net uptake lengths were relatively long; implying that nutrients in these ditches could be transported downstream without significant attenuation. Results from this study indicate that drainage ditches retained P more than NO_3-N. During the study period, periodical increases or decreases in nutrient concentrations were observed along the study reaches; suggesting that these systems were very dynamic in their ability to process nutrient loads. This may be due to the discharge of tile effluents into the receiving waters of the ditches. The study also suggests that nutrient retention efficiency could decrease under high nutrient inputs from tile drains. Incorporation of tile drains into management strategies would improve water quality in agricultural drainage systems and help reduce nutrient delivery to downstream waters.

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